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Dewatering behaviour of water treatment sludges associated with contaminated site remediation in Antarctica

Kathy A. Northcott[a], Ian Snape[b], Peter J. Scales[a], Geoff W. Stevens[a,*]

[a]*Particulate Fluids Processing Centre, Department of Chemical and Biomolecular Engineering, University of Melbourne, Victoria 3010, Australia*
[b]*Human Impacts Research, Australian Antarctic Division, Channel Highway, Kingston, Tasmania 7050, Australia*

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Abstract

Sludge reduction and dewatering is an important aspect of water and waste water treatment. This is especially true in the case of Australia's Antarctic contaminated site remediation program, where the reduction in volume of wastes to be returned to Australia can lead to significant transport and handling cost savings. The dewatering characterisation of water treatment sludges from an Antarctic contaminated site was conducted using a theory of suspension dewatering developed by Buscall, Landman and White. This theory uses fundamental material properties of compressibility and permeability to determine the diffusivity of a suspension. Diffusivity is a useful property that can be used to directly compare the dewaterability of various sludges. In this investigation, several water treatment sludges were collected and characterised in the field to determine the impact of temperature and additives on compressibility, permeability and diffusivity. The Antarctic sludges were found to be less compressible and less permeable than materials such as mineral suspensions and alum water treatment sludges. Compressibility was found to decrease with the addition of powdered coagulation aids such as bentonite and chitosan.

Keywords: Water treatment; Cold regions; Sludge dewatering; Compressional; Rheology

1. Introduction

Dewatering of sludges is of significant economic and environmental importance to the minerals, chemicals and water treatment industries. Disposal of sludges is costly and reduction of sludge volumes through dewatering can provide large cost savings. In addition to the financial incentives, sludge dewatering provides environmental benefits as a result of enhanced water recovery and, in the case of the minerals industry, safer tailings disposal. The dewatering behaviour of suspensions can be complex and is influenced by solids concentration, particle size distribution, liquor density, solid–liquid chemical interactions, solids surface chemistry, chemical additives and many other parameters.

The most common method of sludge dewatering is by removing liquid through the application of a compressive force. Buscall and White (1987) developed a mathematical theory of compressional dewatering involving the use of two physical parameters; compressive yield stress, $P_y(\phi)$, and hindered settling function, $R(\phi)$, to describe the dewaterability of flocculated suspensions. Using these parameters and an appropriate physical model of the equipment being utilised, it is possible to predict the dewatering behaviour of coagulated suspensions. The relationship between these parameters and those used by other researchers for dewaterability characterisation have been compared and contrasted in a recent article (de Kretser et al., 2003).

The use of water treatment technologies has been investigated to mitigate problems with dispersal of contaminants via surface water runoff during Australia's Antarctic contaminated site remediation program. This program aims to recover wastes from and remediate tip sites across Antarctica.

* Corresponding author. Tel.: +61 3 8344 6621; fax: +61 3 8344 4153.
E-mail address: gstevens@unimelb.edu.au (G.W. Stevens).

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The project involves site characterisation, engineering design and design optimisation and characterisation and modelling of the compressional rheological behaviour of sludges at low temperature.

The modelling of dewatering processes, using compressibility and permeability data, is proposed in this work as an essential tool for planning of site remediation at remote sites. It makes it possible to determine the best way to handle and transport sludges produced during site remediation. Whilst this theory and associated dewaterability characterisation techniques have been applied effectively for the minerals and water treatment industries in temperate climates (Green et al., 1996; de Kretser et al., 1997, 1998; Harbour et al., 2001; Usher, 2002), there have been no published applications for cold regions water treatment. Therefore, the aim of this investigation was to specifically look at the impact of low temperature and additives on the dewaterability of coagulated water treatment sludges.

2. Design of a water treatment system for Antarctic applications

An integrated, portable water treatment system has been designed for runoff management during contaminated site clean up in Antarctica. The system comprises a number of treatment stages including particle separation, ion exchange and/or vacuum distillation, all housed in a transportable container. The particle separator section of the water treatment system incorporates coagulation followed by gravitational settling. The settled sludge, containing the majority of contaminants from the treated water, is then drained periodically for disposal.

The water treatment system was initially commissioned and operational tests conducted at the Australian Antarctic Division (AAD), Kingston, Tasmania, prior to transportation to Casey station, Antarctica, for low temperature testing. The water treatment system was operated for a period of 8 weeks during the summer of 2002–2003 to assess water treatment efficiency at low temperature (typically 1–5 °C) and determine settler sludge dewaterability parameters.

3. Experimental section

3.1. Dewatering theory

The Landman theory (Landman et al., 1995) was used to model the sludge dewaterability in the water treatment system, which operates partly as a continuous flow gravity settler and partly as a batch settler. The inputs to this model were the compressibility and hindered settling function behaviour of the sludges.

The extent of dewatering is measured as a compressive yield stress. It is the compressive pressure required to make a networked suspension at a given volume solids fraction, ϕ, yield and consolidate to higher solids concentration. $P_y(\phi)$ increases with ϕ as the number of inter-particle linkages increase, and dewatering occurs until locally $P_y(\phi) = \Delta P$. The compressive yield stress can only be measured at volume solids fractions greater than the gel point, ϕ_g. The gel point is defined as the solids concentration at which particles or flocs are able to form a self-supporting network. It should be noted at this point that although it is common for industry to use weight percent solids as a convention to describe the solids fraction in a particulate suspension, volume fraction is used here since it is this parameter that represents the input to most dewatering models.

The suspension permeability, or resistance to flow through suspensions, is inversely proportional to the hindered settling function, $R(\phi)$. The hindered settling function takes into account the hydrodynamic interaction between particles, or the resistance to flow of liquid through a particle suspension with respect to solids volume fraction, ϕ. The hindered settling function is applicable to the full range of suspension concentrations, as flow can be interpreted as a particle through a liquid or a liquid through a network of particles. So the hindered settling function and hence permeability can be measured as a continuous function of volume solids fraction from dilute solutions through to pastes and filter cakes.

Solids diffusivity is a parameter that combines both the compressibility and hindered settling function and is often used as a measure of dewaterability of a suspension. It is a key input parameter to the modelling of processes such as filtration and centrifugation. There are a large range of models available in this area, many of them empirical. However, phenomenological rheological models that incorporate solids diffusivity have a fundamental basis and use as material property inputs both the solids diffusivity and the compressibility of the sample. This class of model is best suited to flocculated particulate suspensions that develop continuous network structures at low solids volume fractions, usually below 30 vol% solids. This is typical of those studied herein. The key assumption in these models is that the feed concentration to the dewatering device and applied pressure are high enough such that the effects of sedimentation and gravity are insignificant. The solids diffusivity ($D(\phi)$) is described by Landman et al. (1995) and incorporates both the compressive yield stress and the hindered settling function into one relationship such that

$$D(\phi) = \frac{\mathrm{d}P_y(\phi)}{\mathrm{d}\phi} \frac{(1-\phi)^2}{R(\phi)}. \tag{1}$$

Using Eq. (1), dewaterability comparison between suspensions is qualitatively possible, assuming that sample compressibility is similar. Under these circumstances, samples with a higher calculated diffusivity can be considered to be more permeable and hence, more dewaterable. This conclusion is based on the inverse dependence of $D(\phi)$ on $R(\phi)$. A more quantitative assessment can be made by incorporation of the compressibility and diffusivity into an appropriate dewatering model.

3.2. Materials and methods

The ferric chloride coagulant used in all water treatment system tests was laboratory grade $FeCl_3.6H_2O$ AnalR (BDH chemicals). The additives were drilling grade bentonite clay in powder form and chitosan powder, with a high degree of deacetylation, manufactured by Kimitsu Corporation. Tests conducted at the AAD used a sample of water containing colloidal sludge that was residual from the processing of tip materials in the previous year. This sample was of the same origin as feed samples to the water treatment facility in Antarctica but did not contain as much silt and coarse particulate material. All suspensions characterised were sludges formed after dosing ferric chloride to tip overflow waters.

Representative of the process, the entire sludge contents (60 L) of the hopper were emptied into a collection vessel and samples were taken from this vessel. For each sample the process temperature and pH was recorded. For the tests conducted at the AAD in Kingston, three suspensions were characterised. These were a bentonite suspension (AAD FeBe), a bentonite-tip suspension (AAD FeBeTi) and a tip suspension (AAD FeTi). Six suspensions were characterised during the Antarctic tests in the laboratory and the field (low temperature, LT). These were tip suspensions (Ant FeTi, Ant LTFeTi), bentonite-tip suspensions (Ant FeBeTi, Ant LT-FeBeTi) and chitosan-tip suspensions (Ant FeChTi, Ant LT-FeChTi). For all experiments the coagulant used was $FeCl_3$ at a dose of approximately $20\,mg\,L^{-1}$. The dewaterability of the various sludges was determined by a combination of batch settling tests and stepped pressure filtration (de Kretser et al., 2001; Usher et al., 2001; Usher, 2002).

Equilibrium batch settling was used for determination of gel point and for measurement of compressive yield stress at low solids concentrations. The equipment for equilibrium batch settling includes a range of settling cylinders, a ruler for measuring sediment height and sample container. For the purposes of this investigation the "constant initial solids concentration technique" was used. This method was chosen as it has been used in other studies (Aziz, 1998), it requires less sample and does not require multiple sample dilutions or pre-concentrations to set up.

Transient batch settling enabled the determination of the hindered settling function, $R(\phi)$, at low to intermediate solids concentrations. The upper limit for solids concentration in transient batch settling is dictated by the suspension compressive yield stress, which is generally less than 200 Pa (Usher, 2002). The tests are generally conducted in 1–2 l measuring cylinders and settling times are concentration dependent and range from an hour to a day. The test utilises a number of cylinders containing suspensions with solids concentrations above and below the gel point. The initial settling rate is measured by following the suspension–supernatant interface as it travels down the cylinder. The hindered settling function, $R(\phi)$, is calculated from initial solids volume fraction, ϕ_0, initial suspension height, h_0, initial solids settling velocity, u, solid–liquid density difference, $\Delta\rho$, and the compressive yield stress, $P_y(\phi)$, calculated at the initial solids volume fraction via the equilibrium batch settling tests.

A stepped pressure filtration technique has been developed (de Kretser et al., 2001; Usher et al., 2001) that uses one stepped pressure compressibility filtration test and one stepped pressure permeability test to determine $P_y(\phi)$ and $R(\phi)$ at a number of solids volume fractions. The combined results of the two filtration tests can be used to determine $D(\phi)$ for the same range of solids concentrations.

The stepped pressure filtration technique utilises one of two types of pressure filtration apparatus. The first type, known as the "Permacom 3000", is a bench scale filter press, which uses a pneumatic cylinder to compress a sample suspension in a filtration cell. The sample is consolidated by forcing liquid from the sample out through a membrane at the base of the cylinder. The membrane is supported on a sintered disk and perforated plate. The applied pressure is controlled by a computer program, which also measures and logs the position of the piston in the cell via a linear encoder. The "Permacon 3000" can characterise suspensions at pressures of 5–300 kPa and a filtration test can take between 1 and 24 h to complete depending upon the permeability of the suspension.

The second type of filtration apparatus is the air driven filtration rig. The principal of operation for this rig is similar to that of the "Permacom 3000", with the major difference being the use of direct air pressure to compress a sample rather than a piston. Because there is no piston, the linear encoder is absent. Compression is hence measured by the amount of liquid forced from the sample into a container placed upon an electronic balance. The air driven apparatus has a much more simplified design, making it more portable and cheaper to manufacture. It is not suitable for materials at high temperature due to drying out of the sample during testing.

The sludges collected at the AAD were tested using a piston driven filtration rig to a maximum pressure of 300 kPa. The sludges collected in Antarctica were tested using an air driven filtration rig. Due to problems associated with the sample drying out and air by-passing the sample at high pressure, the air driven filtration rig tests were limited to pressures of 100 kPa.

4. Results and discussion

4.1. Sediment characterisation

A previous article gives the complete characterisation of the sediments, which make up the Antarctic sludges tested (Northcott et al., 2003a). To summarise the sediment

Table 1
Operating conditions for sludges collected from the water treatment system

Sample	Process temperature (°C)	Dewatering test temperature (°C)	Process pH
AAD FeBe	13.0	17.5	7.28
AAD FeBeTi	11.5	17.5	7.20
AAD FeTi	13.0	18.0	7.80
Ant FeTi	4.3	16.5	6.78
Ant FeBeTi	2.1	15.5	6.63
Ant FeChTi	5.7	18.0	6.46
Ant LTFeTi	4.1	4.0	6.93
Ant LTFeChTi	1.4	5.0	7.02
Ant LTFeBeTi	0.7	4.0	6.68

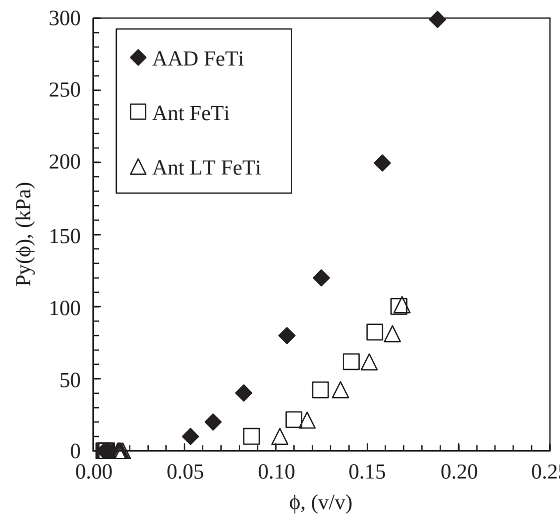

Fig. 1. Ferric-tip sludge compressive yield stress, $P_y(\phi)$, versus solids volume fraction, ϕ.

characterisation experiments, the tip samples are predominantly sandy sediments made up of quartz and feldspar minerals. The tip sediments are in the neutral to slightly alkaline pH range (6.5–8) with higher than normal organic material for Antarctic soils (∼ 10% by mass). Tip sediments contain up to 40% by volume of fine particles in the 0–75 μm size range and tip suspension tests have shown that around 90% of particles readily entrained in suspension are in this size range. The particles found in the tip waters most likely to be associated with colour and turbidity include clays, silts, organic material and minerals. These particles tend to have a negative surface charge and are stable in suspension at and above pH 4.0 (AWWA, 1998).

4.2. *Effect of pH and temperature on particle coagulation and formation of Antarctic sludges*

The measured process temperature and pH for the sludges sampled from the water treatment system are shown in Table 1. Iron cations undergo hydrolysis, forming strong bonds with the oxygen atoms of six water molecules. As a result the oxygen–hydrogen bonds are weakened in the water molecules and hydrogen ions can be released into solution. If the concentration of salt addition is lower than the solubility product of the amorphous hydroxides, a number of hydroxymetal complexes can be formed, from monomeric through to giant cation molecules. Fe is amphoteric, with both cationic and anionic complexes formed. If the solubility product is exceeded for $Fe(OH)_3$, a series of hydrolysis reactions result in amorphous metal hydroxide precipitates. In practice, the dosage of Fe^{3+} is always high enough to exceed the solubility of the hydroxides (Faust and Aly, 1996).

The pH of the system is important for the performance of metal salts as coagulants and to minimise residual metal ions in treated water. At low pH soluble hydrolysis products and aquo metal ions are formed, whilst at high pH soluble $Fe(OH)_4^-$ is produced. Ferric hydroxide minimum solubility occurs at around a pH of 8. In general the pH of minimum solubility increases with a decrease in temperature. For industrial processes, the optimum pH of coagulation varies but is generally within a specified range. Ferric salts are most effective between pH 5.0 and 8.5, with a typical operating pH of 7.5 (AWWA, 1998). In this investigation, the operating pH was generally lower than a typical operating pH for a coagulation system utilising ferric chloride. However Table 1 shows that the water treatment system consistently operated in the normal 5.0–8.5 operating range for a ferric chloride system.

Temperature is an important consideration for coagulation and flocculation. Water viscosity increases with decreasing temperature and as a result the rate of floc formation tends to decrease. The impact of low temperature is greatest on dilute suspensions. However, it is believed that chemical factors are more significant than fluid motion effects at low temperature (Kang and Cleasby, 1995). Temperature affects the solubility of metal hydroxide precipitates and the rate of metal hydrolysis and metal hydroxide precipitation is reduced at low temperature. A detailed investigation of the coagulation and flocculation conditions of the water treatment system is provided in another paper (Northcott et al., 2005). In summary, this investigation found that flowrate, coagulant dose, pH and raw water turbidity had a direct influence on final treated water turbidity, whilst bentonite and chitosan addition appeared to have little effect on water treatment. Temperature of coagulation was found to have no measurable impact on floc formation, however, the increased water viscosity at low temperature required careful consideration of the flowrate through the settler to achieve adequate particle settling.

4.3. *Dewatering characteristics of ferric sludges*

The compressive yield stress of the AAD and Antarctic ferric-tip sludges are shown in Fig. 1. These sludges were coagulated using $FeCl_3$ and no other additives were used, other

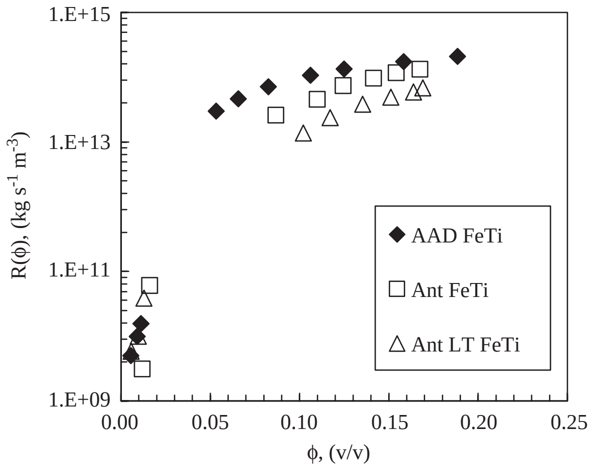

Fig. 2. The hindered settling function, $R(\phi)$, of ferric-tip sludge versus solids volume fraction, ϕ, precipitated in the presence of ferric chloride.

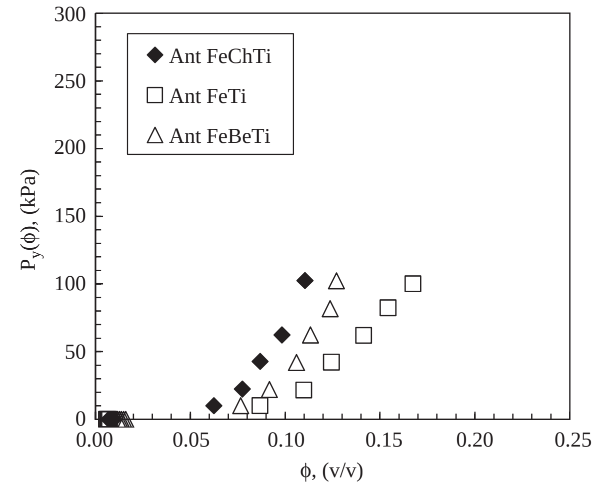

Fig. 3. Comparison of compressive yield stress, $P_y(\phi)$, versus solids volume fraction, ϕ, for Antarctic water treatment sludges containing different compositions of coagulant and coagulation aids.

than NaOH for pH adjustment. The two Antarctic ferric-tip sludges have similar compressibility characteristics despite the difference in dewaterability characterisation temperature (see Table 1). The AAD ferric-tip sludge is not as compressible as the Antarctic sludges, achieving a solids volume fraction of around 0.05 at 10 kPa compared to approximately 0.1 for the Antarctic sludges, at the same pressures.

We attribute difference in compressibility to two possible phenomena. The process temperature during coagulation for the AAD tests was around 13 °C, whilst for the Antarctic tests coagulation occurred at 1–6 °C. The difference in temperature can have an impact on the structure and strength of the aggregates produced, resulting in a change in suspension compressibility. It is more likely, based upon the results of coagulation and flocculation analysis (Northcott et al., 2005), the difference in results are due to the chemical nature of the particles and colloids in suspension. The suspensions collected directly from the Thala Valley tip site contained less weathered particles and produced less dissolved metals in solution than the processed sample used in tests at the AAD in Kingston, Tasmania. Despite the different process routes, differences in the physical nature of the feed material were not significant and the majority of the sludge produced was made up of ferric hydroxide precipitates.

Overall it was found that ferric sludges, compared to mineral suspensions such as red mud and alumina (Usher, 2002; Harbour et al., 2004) are highly incompressible achieving final solids fractions of less than 20% v/v at pressures up to 100 kPa. This indicates that in order to achieve high solids via filtration, extremely high pressures would be required to process the sludge.

The hindered settling function with respect to solids volume fraction for the case of ferric coagulant added to a tip sample is shown in Fig. 2. The hindered settling function is inversely proportional to the permeability of a suspension, hence the larger the value, the lower the permeability. The hindered settling function changes by several orders of magnitude over the solids range. The data are calculated from batch settling at low solids and high solids pressure filtration at high solids.

There is a clear difference in the hindered settling function and hence permeability between the three ferric-tip suspensions. The difference is moderate at intermediate solids and the data appear to converge at higher solids (Fig. 2). Overall it appears that the nature of the ferric hydroxide precipitates produced by the addition of ferric coagulant has the most significant effect on suspension permeability, particularly at higher solids fractions. The similarity of the data between the Antarctic samples at normal and low temperatures suggests that dewatering temperature is less important than the physical and chemical properties of the sludge. The data show that the AAD sample to be less permeable and from Fig. 1, less compressible that the fresh Antarctic sample. The hindered settling function of the ferric sludges, at the solids fractions achieved during high pressure filtration, was found to be orders of magnitude higher than that found at similar solids fractions in red mud and alumina suspensions (Aziz, 1998; Aziz et al., 2000; Usher, 2002). Hence ferric suspensions can be classed as highly impermeable, with a large resistance to water flow though the filter cake.

4.4. Bentonite and chitosan

Chitosan and bentonite treated suspensions were tested in the water treatment system, to assess their potential as flocculation aids (Northcott et al., 2005). Comparison of three of the Antarctic tip suspensions; Fe only, Fe and bentonite and Fe and chitosan addition shows that the ferric-tip suspension is more compressible than the suspensions with bentonite and chitosan as additives (see Fig. 3). Hence at the same pressure, Fe-only achieves a higher solids fraction. Considering this and the fact that addition of bentonite and

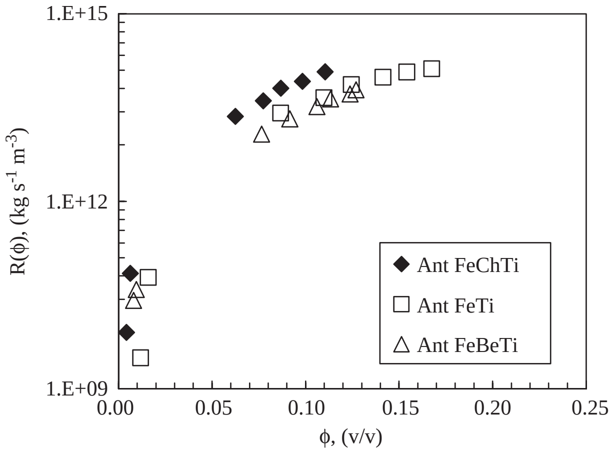

Fig. 4. Comparison of hindered settling function, $R(\phi)$, versus solids volume fraction, ϕ, for Antarctic water treatment sludges of different coagulant and coagulation aid compositions.

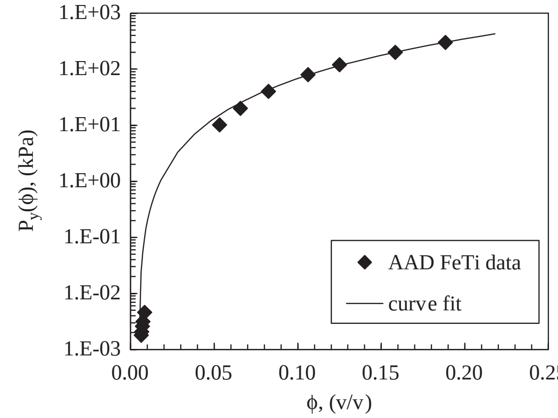

Fig. 5. Compressive yield stress, $P_y(\phi)$, curve fit for AAD ferric-tip sample (fitting parameters $p_a = 46.06$, $p_b = -5.12$, $p_m = 5.2\text{E} - 12$, $p_n = 0.049$).

chitosan increases the suspended solids loading and hence the amount of settled sludge, the use of additives in this case had a detrimental effect on the extent of dewatering.

Comparison of three Antarctic samples treated with Fe, Fe–bentonite and Fe–chitosan, show that the ferric-tip and ferric–bentonite-tip sludges have very similar permeability characteristics, whilst the sample treated with chitosan is less permeable (i.e., a higher hindered settling function, see Fig. 4). The difference in permeability between suspensions treated with bentonite and chitosan could be due to the nature of the particles of each material. Bentonite is a swelling clay, whereas chitosan is a granular material. Hence the addition of chitosan may produce a floc structure with different permeability characteristics than for the bentonite or ferric-only case.

Overall there is no improvement in compressibility or permeability characteristics of tip suspensions with the addition of bentonite or chitosan. It is concluded on this basis that if the use of additives such as bentonite and chitosan is going to be of benefit to the overall water treatment process it would only be either from improved water treatment system stability or dissolved metals removal (Northcott et al., 2003b). Indeed, chitosan and bentonite have high metals adsorption capabilities and there is potential for a combined suspended and dissolved heavy metals removal. The combined use of ferric chloride with bentonite or chitosan may be a low cost environmentally friendly alternative for removing both suspended and dissolved heavy metals from contaminated waters. Work is currently under way to investigate the combined coagulation aid and ion exchange potential of bentonite and chitosan (Northcott et al., 2003b).

4.5. Determination of diffusivity of ferric sludges

The diffusivity of coagulated suspensions is determined by fitting appropriate functional forms to compressibility and hindered settling data followed by input into Eq. (1). The diffusivity is a material property, which along with the compressibility, can be used to characterise the dewatering behaviour of a suspension. It can be used as a qualitative comparison of dewaterability of different suspensions or can be used to quantitatively predict time of filtration for various pressure filtration processes.

The functional form used to fit the compressive yield stress data over the full range of solids was as follows (Usher, 2002):

$$P_y(\phi) = \left(1 - \left(\frac{\phi_g}{\phi}\right)^{p_m}\right) \mathrm{e}^{(p_a \phi^{p_n} + p_b)}, \tag{2}$$

where p_a, p_b, p_m and p_n are fitting parameters. This equation is fitted to the compressive yield stress data by using a least squares minimisation technique. Whilst Eq. (2) is complex, it has been shown to give an accurate and smooth curve fit to experimental compressibility data of suspensions over the full range of solids fractions. Compressive yield stress data at solids close to the gel point were determined via equilibrium batch settling, whilst high solids data were found using pressure filtration. Fig. 5 is the curve fit for the AAD ferric-tip sludge sample compressive yield stress data. The fitting parameters calculated for all sludge samples tested are given in Table 2.

The functional form used for hindered settling function curve fitting is given by (Usher, 2002)

$$R(\phi) = r_a \phi^{r_n} + r_b. \tag{3}$$

Eq. (3) is a simple power law functional form where r_a, r_b and r_n are fitting parameters. Low solids data around the gel point were determined via batch settling and high solids data were measured by pressure filtration. Fig. 6 is an example of the hindered settling function curve fit for the AAD ferric-tip sample. There is some scatter around the $R(\phi)$ curve fit, however, as the hindered settling function spans several orders of magnitude the significance of the scatter is reduced.

Table 2
Fitting parameter data for ferric sludge samples

Sample	p_a	p_b	p_m	p_n	r_a	r_n	r_b
AAD FeBe	43.73	−4.47	9.6E − 12	0.044	8.2E+16	3.35	0.00
AAD FeBeTi	44.88	−5.33	1.2E − 11	0.049	2.0E+16	2.96	0.00
AAD FeTi	46.06	−5.11	5.1E − 12	0.049	2.0E+17	3.43	0.00
Ant FeTi	45.45	−4.69	1.7E − 11	0.074	1.5E+17	3.72	0.00
Ant FeBeTi	40.00	−5.00	6.2E − 7	0.14	2.2E+16	2.87	1.00
Ant FeChTi	46.98	−5.00	1.9E − 10	0.099	1.3E+17	3.15	1.00
Ant LTFeTi	44.70	−5.00	3.2E − 10	0.11	1.09E+16	2.88	1.00
Ant LTFeChTi	55.91	26.6	4.2E − 11	0.89	6.83E+17	3.80	1.00
Ant LTFeBeTi	47.14	−5.63	7.8E − 12	0.070	5.57E+15	2.75	0.00

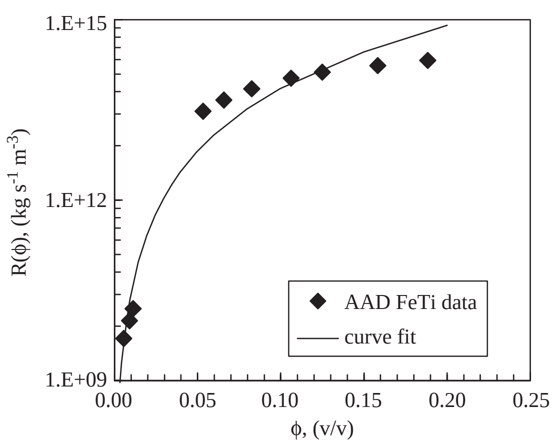

Fig. 6. Hindered settling function, $R(\phi)$, curve fit for AAD ferric-tip sample (fitting parameters $r_a = 1.98\text{E}+17$, $r_n = 3.43$).

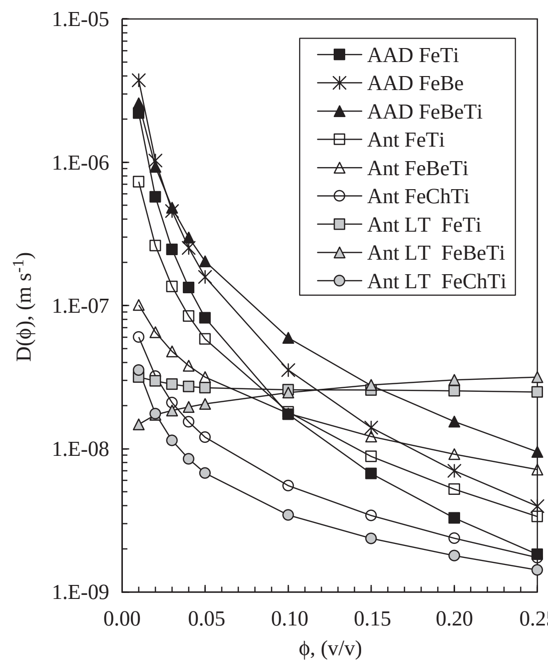

Fig. 7. Solids diffusivity, $D(\phi)$, of ferric sludges calculated using curve fitting method.

The hindered settling function fitting parameter data for all sludge samples is shown in Table 2.

The curve fits for $P_y(\phi)$ and $R(\phi)$ are used in Eq. (1) to calculate diffusivity for suspensions over the full range of solids. Fig. 7 shows the calculated diffusivities for the nine suspensions analysed. Analysis of the diffusivity of a range of sludges shows that for mineral type suspensions, $D(\phi)$ is usually an increasing function of solids concentration in the 0–100 kPa pressure range. Nonetheless, the functional form must show a peak at some point and many lower permeability materials such as potable water treatment sludges show this behaviour (Harbour et al., 2004). It is not surprising therefore that many of the samples in Fig. 7 show a decreasing trend in $D(\phi)$ with increasing solids concentration.

The AAD sludge samples have the highest diffusivities at low solids (up to a solids fraction of 0.05), followed by the Antarctic ferric-tip sample. At higher solids, the AAD sample diffusivities fall into the same region as for the Antarctic sample diffusivities. The Antarctic samples (with the exception of the Ant FeTi sample) have a flatter and more consistent diffusivity profile across the solids range. The Antarctic low temperature ferric-tip diffusivity profile is almost constant, whilst the Antarctic low temperature sludge with bentonite addition has a profile with a trend of increasing $D(\phi)$ with increasing solids.

The trend of decreasing diffusivity with increasing solids is not unexpected but as indicated previously, is usually only seen for poorly dewaterable materials. The diffusivity for mineral suspensions such as red mud (Hulston and Scales, 2001) and alumina (Aziz, 1998) has been found to increase with increasing ϕ. Many organic sludges that characteristically exhibit long dewatering times to achieve final solids at a given applied pressure also show indications of a decreasing diffusivity trend with increasing solids concentration (Stickland et al., 2005).

4.6. *Application of sludge dewatering theory and further work*

Pressure filtration is an energy intensive process, generally requiring expensive and mechanically complex equipment.

The results of the sludge characterisation experiments described here have shown that there is little benefit using pressure filtration for dewatering of ferric sludges, unless very high pressures are used. The Antarctic sludge dewatering characterisation experiments found that the ferric sludges tested were typically incompressible and impermeable in comparison to other suspensions (Aziz et al., 2000; Hulston and Scales, 2001; Usher, 2002), achieving solids volume fractions of around 20% v/v at a pressure of 100 kPa.

From the results of dewaterability characterisation, along with consideration of the constraints of working in remote cold regions sites, two dewatering processes were investigated. Batch settling analysis uses a sludge consolidation model based on compressibility and hindered settling function data. This model can be used to determine the time taken for a suspension to settle to a final solids volume fraction. This is particularly useful for planning for site remediation at remote sites. If the rate and extent of settling of the sludges is known it is possible to determine the number of containers required for storage and how often drums can be decanted and extra material added. The second process is freeze–thaw, where the sludge is transformed into granular particles with high solids content. Freeze–thaw has been investigated previously for the dewatering of alum water treatment sludges (Martel and Diener, 1991; Martel et al., 1998). The practical application of these techniques to Antarctic sludge dewatering will be covered in detail elsewhere (Northcott et al., 2005) and are part of our ongoing research.

5. Conclusions

The theory of Buscall and White was used to characterise sludges taken from a particle separation system in Antarctica. The theory has been used extensively to describe dewatering behaviour of mineral slurries and to model solid–liquid separation processes, however, it has not been previously applied to cold regions water treatment.

The dewatering experiments found that ferric sludges, compared to mineral suspensions such as red mud and alumina, are highly incompressible achieving final solids fractions of less than 20% v/v at pressures up to 100 kPa. The hindered settling behaviour of the suspensions is consistent for all materials tested. The hindered settling of the ferric sludges is orders of magnitude higher than that found at similar solids fractions in red mud and alumina suspensions. This indicates that the ferric suspensions are also highly impermeable, with a large resistance to water flow though the filter cake. Comparison of sludges dewatered at room temperature and at low temperature found that dewatering temperature is less important than the physical and chemical properties of the sludge.

In general, the ferric sludge samples show decreasing solids diffusivity with increasing solids fraction. This trend is unusual, as the normal diffusivity trend for mineral suspensions is for increasing diffusivity with increasing solids fraction. Another material, which has a decreasing diffusivity with increasing solids characteristic, is sewage sludge, which is known to be highly impermeable and characteristically exhibits long dewatering times to achieve final solids at a given applied pressure. It is possible that this similar behaviour can be exploited by designing a dewatering system that is optimal for both Antarctic tip leachate and sewage sludges. The cost benefit of sludge reduction in remote cold regions means that this research has immediate application.

Notation

$D(\phi)$	diffusivity, m s^{-1}
h_0	initial suspension height, m
p_a	compressibility curve fitting parameter, dimensionless
p_b	compressibility curve fitting parameter, dimensionless
p_m	compressibility curve fitting parameter, dimensionless
p_n	compressibility curve fitting parameter, dimensionless
$P_y(\phi)$	compressive yield stress, kPa
r_a	hindered settling function curve fitting parameter, dimensionless
r_b	hindered settling function curve fitting parameter, dimensionless
r_n	hindered settling function curve fitting parameter, dimensionless
$R(\phi)$	hindered settling function, kg s^{-1} m^{-3}
u	solids settling velocity, m s^{-1}

Greek letters

ΔP	change in compressive pressure, kPa
$\Delta\rho$	solid–liquid density difference, kg m^{-3}
ϕ	solids volume fraction of suspension, dimensionless
ϕ_g	gel point of suspension, dimensionless
ϕ_0	initial solids volume fraction, dimensionless

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